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Routes to achieving sustainable intensification in simulated dairy farms - the importance of production efficiency and complimentary land uses

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ABSTRACT

1. Sustainable intensification (SI) is a global challenge, aiming to increase food production whilst conserving biodiversity and ecosystem services. This is contrary to the observed trend of agricultural intensification degrading environmental quality. We developed a framework integrating animal nutrition, crop yields, and biodiversity modelling to explore SI potential in multiple model dairy farming systems through varying crop composition to provide cattle feed rations. We then identified key drivers of biodiversity gain that may be applicable at a wider scale.
2. We developed multiple feed rations to meet the nutritional demands of a high-yielding, housed dairy herd. The land area required varied due to productivity and nutritional differences between crops, generating spare land. We used published biodiversity models to compare alpha- and beta-diversity of spiders and plants across 36 scenarios that used the spare land in different ways, for either biodiversity maximisation or additional production.
3. Alpha and beta-diversity for both taxa was greatest in scenarios that maximised spare land and utilised this for species-rich extensive grassland. However, commensurate biodiversity gains for plant alpha-diversity, and spider and plant beta-diversity (respectively 100%, 76% and 86% gain relative to that optimal scenario) were achievable when spare land was used for additional crop production.
4. Maximising compositional heterogeneity and adding complementary productive land uses to spared land were key to increasing production and beta-diversity, while adding species-rich productive land uses drove increasing production and alpha-diversity.
5. *Synthesis and applications.* This study indicates the potential for the sustainable intensification (SI) of dairy farming systems through the manipulation of feed rations to increase land efficiency and spare land, which could then be used to enhance production and biodiversity. The optimum land composition depends on target goal(s) (e.g. maximising

production and/or biodiversity). Greatest ‘win-wins’ can be achieved by making land cover more diverse and selecting crops that complement each other in the species they support; highlighting the important role of heterogeneity in the crop matrix. Our study provides a framework that integrates agricultural production efficiency and biodiversity modelling to explore potential routes to achieve SI goals.

KEYWORDS

agriculture, biodiversity, dairy, ecological intensification, farming, heterogeneity, land use, sustainable intensification

INTRODUCTION

The challenge of protecting biodiversity and ecosystem services in the face of increasing food production has led to the call for sustainable intensification (SI) of agricultural land; that is, the simultaneous increase of food production and reduction of environmental pressure (Tilman *et al.* 2011; Garnett *et al.* 2013). This is now a key strategic objective of the UN's Food and Agriculture Organisation (FAO 2015). A premise of SI is that increased production should be achieved on existing farmland, because clearance of other habitats for agriculture creates greater environmental degradation and biodiversity loss (Tilman *et al.* 2011; Garnett *et al.* 2013). Thus, a question central to SI is whether we can increase production of existing land while maintaining or improving its biodiversity.

Livestock production and biodiversity

Demand for meat and dairy products have increased substantially, with global cattle stock increasing by 57% between 1961 and 2016 (FAOSTAT 2017). Grassland management has been intensified, changing the functional composition of vegetation, with associated global biodiversity declines being observed across taxa (Plantureux, Peeters & McCracken 2005). Functional and/or taxonomic changes to vegetation composition have benefitted some ecosystem services (e.g. food provisioning / nutrient cycling services), at the cost of others (e.g. regulating, cultural and biodiversity services) (Allan *et al.* 2015).

To satisfy the nutritional demands (i.e. fat, carbohydrate, protein) of high-yielding dairy cows (≥40 L milk d^{-1} $animal^{-1}$), feed rations comprise of multiple crops (Toma *et al.* 2013). Variation in the composition of animal feeds and in-crop productivity produces large differences in the amount and composition of land required to fulfil the dietary needs of a given herd at a target milk yield. In the context of SI, this creates a dual opportunity: (1) if land-use composition can be modified to increase

81 production, modifications may be steered towards productive land compositions that maintain, or
82 even improve, biodiversity; (2) if efficiency improvements mean land-use composition can be
83 modified to produce spare land, whilst still maintaining or increasing production, this land might be
84 used for high biodiversity land-uses. Grassland simulations indicate heterogeneity of management
85 intensity can be varied to simultaneously increase arthropod populations and food production
86 (Simons & Weisser 2017). However, this has not been explored in mixed land-use systems where
87 there may be greater scope to manipulate crop composition to achieve multiple benefits.
88
89 Our study integrates animal nutrition, crop productivity, and biodiversity modelling to explore a
90 range of land-use scenarios that meet the feed requirements of a model dairy system to determine
91 the possibility of increasing food production whilst maintaining or enhancing biodiversity. We
92 developed a range of scenarios and manipulated the composition of land-covers to determine where
93 'win wins' could be achieved and to identify key drivers of production and/or biodiversity gains. All
94 scenarios provided feed for a herd of at least 100 permanently housed dairy cattle but differed in
95 terms of their land-use composition and thus land-efficiency (Fig. 1). More land-efficient scenarios
96 generated up to 18% 'spare land', which could then be used for additional production, or allocated
97 to a biodiversity-rich habitat (i.e. extensive grassland). Extending published land-use/biodiversity
98 models, we estimated indices for scenarios' alpha and beta-diversity for two functional groups
99 (plants and epigeal spiders) where extensive biodiversity data were available (Downie *et al.* 1999;
100 Wilson *et al.* 2003). We hypothesised that production of the system could be maintained or even
101 improved, whilst also improving or maintaining diversity of plants and spiders. We thus explore the
102 potential for SI within a dairy system and discuss implications for achieving SI goals more widely.
103
104 **MATERIALS AND METHODS**
105 **Land-use scenarios**

Our modelling process explored a 'home-grown feeds' dairy system, where nutritional requirements of permanently-housed dairy cows are grown on the farm (Roberts & March 2014). We used an industry-standard livestock nutrition feeding model, FeedByte (Scottish Agricultural College 2006), to design alternative feed rations for a model herd of 100 Holstein-Friesian cows with a target milk yield of 9,500 L yr^{-1} (equivalent to 40 L of milk cow^{-1} day^{-1} including an unproductive period). This commercially-used software uses least-cost diet formulation and linear-programming and is used by industry and researchers to design real feed rations sufficient for livestock herds (Chagunda, Flockhart & Roberts 2010). Rations needed to comprise of grass silage, barley, wheat, oilseed rape and fodder beet as comprehensive biodiversity data existed for these crops and they varied with respect to the availability of specific nutrients. Each crop alone would not provide the combination of nutrients (e.g. proteins, carbohydrates or fats) required for high-yielding cattle. For each ration, we used productivity estimates using both fresh-weight and dry-matter (DM) yields (kg ha^{-1}) as well as nutritive value estimates (g kg^{-1}) for the constituting crops (Supporting Information Table S1) to calculate the land area required. Due to differences in crop yields and nutritive values, the amount of land required to meet the herds' nutritional requirements differed substantially between rations. The feed ration requiring the most land for our target milk yield and herd size (i.e. the least land-efficient ration) was designated the 'baseline scenario' (103.8 ha) (Fig. 1a). Our most land-efficient feed ration provided the same milk yield and herd size on less land (88.0 ha). Consequently through manipulating the quantities of different crops in the feed ration we could generate up to 15.8 ha (18%) of spare land relative to the baseline. This generated our maximum spare land test-system (Fig. 1b).

To compare a variety of scenarios where feed rations are met via different land areas and compositions, we generated five intermediate test-systems varying in land-efficiency between the baseline scenario and the max. spare land test-system (1%, 6%, 8%, 12% and 15% spare land c.f. the

baseline). The result was six test-systems (Fig. 1b) containing spare land; the choice of six allowed generation of 36 'spare-land scenarios', considered a suitable sample size for exploring drivers of biodiversity change across scenarios. We generated spare-land scenarios by replacing the spare land component of each test-system (Fig. 1b) with one of six land uses (grass silage, barley, wheat, oilseed rape, fodder beet or extensive grassland) (Fig. 1d). Spare land scenarios thus differed in both the amount of spare land and the land use replacing the spare land component. Spare land scenarios could be divided into:

(1) 'additional production' scenarios where spare land was used to grow more of one of the productive crops (silage, wheat, barley, oilseed rape or fodder beet; Fig. 1d i).

(2) 'no additional production' scenarios where spare land was designated to extensive grassland (Fig. 1d ii). Of the land uses available in our empirical datasets, we selected extensive grassland to represent a high-biodiversity, low/non-productive land use with no improvement or cutting regime.

Sparing land from production and designating to biodiversity-rich habitats, as in our 'no additional production' scenarios, is a frequent focus of agri-environment prescriptions (e.g. fallow, floristically diverse field margins). However, the original datasets we used (see below) did not contain data from such habitats so we used data from extensively grazed, semi-natural grassland (e.g. calcifugous and *Juncus*-dominated) to represent a low-production, high biodiversity land use as our alternative strategy to increasing production. These are not directly equivalent to newly created agri-environment habitats, since environmental (e.g. topography, altitude, soil, climate) and socio-ecological (grazing regimes and underlying productivity of the land) constraints would have historically prevented intensification on such areas with long-term extensive management resulting

155 in unique, species-rich, communities (Downie *et al.* 1999; Wilson *et al.* 2003; Cole *et al.* 2005). We
156 discuss the implications of this for our findings below.
157
158 We also explored an additional scenario where the desired outcome was to maximise milk
159 production ('max. herd size': Fig. 1c). This was achieved by scaling up the land composition of the
160 most land-efficient test-system, which had 18% spare land ('max. spare land'; Fig. 1b), to occupy the
161 area of the baseline scenario resulting in an increase from 100 to 118 cattle and thus increasing milk
162 production by 18%. Thus we generated 38 scenarios in total: the baseline scenario (Fig. 1a), the
163 max. herd size scenario (Fig. 1c), and the 36 spare land scenarios (Fig. 1d).
164
165 **Alpha diversity, beta-diversity, and additional production estimates**
166 For each of the 36 spare land scenarios (Fig. 1d) and the max. herd size scenario (Fig. 1c), we
167 calculated indices of alpha and beta-diversity for plants and spiders, and compared these with the
168 baseline scenario. This analysis framework is illustrated in Fig. 2. Diversity indices were generated
169 from published biodiversity models derived from field studies measuring plant and spider species
170 richness in the agricultural land-covers in our system (Downie *et al.* 1999; Wilson *et al.* 2003).
171
172 Data had been collected in nine geographical locations in Scotland ranging from intensive arable and
173 grassland landscapes, mixed farming and crofting. Vegetation surveys were conducted over three
174 years (1995-1997) at 87 sites with data being collected from permanent 10×10 m quadrats on three
175 occasions during peak growing season (June-September) (Wilson *et al.* 2003). Epigeal spiders were
176 surveyed over two years (1996-1997) at a total of 71 sites using a 16 m transect of nine pitfall traps
177 (75 mm diameter and 100 mm deep). Pitfall trapping was conducted May-September with contents
178 collected monthly (Downie *et al.* 1999).
179

Plants and spiders are key components of agricultural ecosystems, strongly driven by land management (Wilson *et al.* 2003; Batáry *et al.* 2012). Plants are key drivers of invertebrate biodiversity (Symstad, Siemann & Haarstad 2000) and provide shelter and breeding sites for many species, while spiders are important polyphagous predators contributing to natural pest control (Herzog *et al.* 2013). Alongside wild bees and earthworms, they provide appropriate and robust metrics for measuring agricultural biodiversity (Herzog *et al.* 2013).

To generate alpha-diversity estimates for spiders and plants we used species richness (S) models fitted by Downie *et al.* (1999) and Wilson *et al.* (2003). These models were developed to explore drivers of field-scale S in a range of land uses. The process we used to generate an alpha-diversity index (ADI) for each scenario is described in detail in Appendix S2. Briefly, the ADI for each scenario is defined as the mean predicted field-scale S across land-uses, weighted by the proportional area of each land-use. Weighted mean S is a recognised metric to explore impacts of agricultural land-use composition on biodiversity (e.g. Hiron *et al.* 2015).

To generate beta-diversity indices we used data describing the assemblage structure of our target groups collected by Downie *et al.* (1999) and Wilson *et al.* (2003). Beta-diversity here describes the extent of variation of species assemblages between different land-uses. Raw species data for the models considered in the ADI analyses were not available. However, detrended correspondence analysis (DCA) axes scores derived from the raw data were available (Murphy *et al.* 1998). The distance between two sites in a DCA ordination provides a measure of similarity in species composition, with smaller distances indicating greater similarity (Smol *et al.* 2005). The process we used to generate beta-diversity indices (BDI) for each scenario is described in detail in Appendix S2. Briefly, the BDI for each scenario is defined as the median DCA distance between random pairs of

land-uses within each scenario, weighted by the proportional area of each land-use. We provide R code for the estimation of ADI and BDI in Appendix S3

While each spare-land scenario (Fig. 1d) maintained our target milk yield, a scenario's 'value' can be modified relative to the baseline scenario in two main ways: (i) biodiversity value, indicated by ADI and BDI; and (ii) production value, which can be increased when the spare-land component is used for additional production. To determine how spider and plant ADI and BDI changed relative to the baseline scenario we subtracted the baseline ADI and BDI values from those calculated for each scenario, the results termed Δ_{ADI} and Δ_{BDI} (Fig. 2).

To allow us to simultaneously explore both the change in biodiversity and production for additional production scenarios, we estimated the production capacity of the spare land component as metabolisable energy ('ME', MJ kg^{-1} DM) to standardise this benefit across different crops. ME provided a more direct measure of production capacity than monetary value of crops, which is dependent on a range of additional market factors. We derived ME of the spare land component for each additional production scenario by multiplying industry estimates of ME by yield (kg DM ha^{-1}) (SAC Consulting 2014) and area (ha). The baseline scenario did not produce any additional energy on top of feeding the herd of 100 cattle, and consequently the calculated ME value reflects change relative to the baseline (i.e. Δ_{ME}).

Drivers of relative alpha and beta-diversity

To investigate which factors drove the magnitude of Δ_{ADI} or Δ_{BDI} for each taxa, we generated four candidate sets of general linear mixed models, with Δ_{ADI} or Δ_{BDI} for plants and spiders as the response variables. Each spare land scenario provided a single estimate of Δ_{ADI} and Δ_{BDI} (Fig. 2) so the sample size for the analysis was 36. We included test-system identity (Fig. 1b), and land-use

229 constituting the spare-land component (Fig. 1d) as random effects. We describe the fixed effects
230 included in our models in Table 1.
231
232 For each taxa (plant/spider) and response variable (Δ_{ADI} /Δ_{BDI}) we compared models using a small-
233 sample Akaike's Information Criterion (AICc) and Akaike's weights (w_i) (Johnson & Omland 2004).
234 We included several multivariate models and the appropriate null model. We restricted inference on
235 the key drivers of Δ_{ADI} or Δ_{BDI} to examining the smallest set of models containing a summed $w_i \geq 0.90$.
236
237 **RESULTS**
238 **Impact of land use on alpha and beta diversity**
239 Predicted estimates of S for spiders and plants from the alpha-diversity models for individual land-
240 uses are shown in Figs. 3a-b. Predictions of spider S were as low as 11.7 (lower 95% confidence limit
241 [CL_{95}] for silage) and as high as 41.9 (upper CL_{95} for extensive grassland). For plants, values ranged
242 from 6.1 species (lower CL_{95} for barley) to 29.6 (upper CL_{95} for extensive grassland). These were
243 within the observed field-scale ranges from the original data-sets of 10-56 spider species (Downie *et*
244 *al.* 1999) and 5-57 plant species (Wilson *et al.* 2003).
245
246 Assemblage structure of spider and plant communities was most similar in oilseed rape and cereals
247 (i.e. barley and wheat) with communities in these crops showing the greatest disparity with those in
248 extensive grasslands and fodder beet (Figs. 3c-d).
249
250 **Comparison of alpha and beta diversity between scenarios**
251 The baseline scenario had a predicted ADI (the mean field-scale S across land-uses, weighted by the
252 proportional area of each land-use) of 19.8 (CL_{95} 17.8-22.8) spider species and 13.4 (CL_{95} 11.9-15.2)
253 plant species. Δ_{ADI} and Δ_{BDI} for each taxa and scenario are shown in Fig. 4. In all cases, 'no additional

production' scenarios with maximum spare land gave the highest biodiversity benefit (labelled 'NAP' in Fig. 4). However, for both taxa Δ_{BDI}, and for plant Δ_{ADI}, additional production scenarios existed that generated similar biodiversity gains to the latter (optimal scenarios labelled 'AP' in Fig. 4). For spiders, additional production scenarios did not enhance ADI due to the baseline scenario having a relatively high ADI (Fig. 4a).

Drivers of change in alpha and beta-diversity

For both spider and plant Δ_{ADI}, a single model was in the confidence set (Table 2). This contained the estimated S of the added land-use, the area of spare land, and their interaction (Figs. 5a-b). Adding a land-use with a comparatively high estimated S to the spare-land component increased Δ_{ADI}, and this interacted positively with amount of spare land.

For spider Δ_{BDI}, two models were in the confidence set (Table 2), containing the mean DCA distance for the land-use being added, the land-use evenness (w_i = 0.87) or heterogeneity (w_i = 0.08), and their interaction. A higher mean DCA distance of the land-use being added increased Δ_{BDI}, but only where land-use evenness was also increased (Fig. 5c). For plant Δ_{BDI}, four models were in the confidence set (Table 2). The two best models contained either the land-use heterogeneity (w_i = 0.71) or evenness (w_i = 0.13), showing positive relationships (Fig. 5d).

'Value' of spare-land scenarios

For both Δ_{ADI} (Fig. 6a) and Δ_{BDI} (Fig. 6b) utilising spare land as extensive grassland delivered the optimal biodiversity gain across taxa, but generated no Δ_{ME} gain. For Δ_{BDI}, not only was a max. spare land + fodder beet scenario able to achieve gains almost commensurate with the best no additional production scenario for both spiders and plants, but it did so while achieving the highest Δ_{ME} (Fig. 6b).

For Δ_{ADI}, the relatively high ADI value for spiders in the baseline scenario resulted in a complex picture (Fig. 6a). All additional production scenarios showed a reduced ADI for spiders, but increased ADI for plants. Some added fodder beet scenarios achieved relatively high plant Δ_{ADI} and moderate Δ_{ME} gains but at a cost to spider Δ_{ADI} values (albeit with spider losses lower in magnitude than plant gains). Some added silage scenarios achieved greater plant Δ_{ADI} gains than added fodder beet (equal to plant Δ_{ADI} gain in the best no additional production scenario) but at a cost to spider Δ_{ADI} and a lower Δ_{ME}, indicating trade-offs between potential gains.

DISCUSSION

Sustainable intensification outcomes

Simultaneous gains in production and biodiversity were simulated in our model dairy system illustrating the potential for SI. We achieved the biggest simultaneous gains in beta-diversity and production by increasing the system's efficiency to feed the dairy herd using a smaller area of land, and allocating the spare land generated to a productive crop that complemented those already present in the system with respect to the species supported. For plant alpha-diversity, we achieved highest gains in production and species richness by allocating the spare land to a species-rich, productive land use.

Our simulations challenge the well-observed negative relationship between productivity and biodiversity in agriculture (Krebs *et al.* 1999). SI has been demonstrated in small farms in developing countries using a combination of approaches including integrated plant nutrient systems, no-till/conservation agriculture and integrated pest management (Chappell & LaValle 2011). Evidence of SI in highly-productive agricultural systems is less well documented with gains in biodiversity typically being accompanied by yield losses (Gabriel *et al.* 2010; Firbank *et al.* 2013). SI appears

particularly challenging in intensive dairy farms - a study exploring innovative management practices found dairy farms notable in their failure to achieve simultaneous production and biodiversity gains (Firbank *et al.* 2013). Our results highlight a potential route to SI in home-grown dairy systems.

With increasing pressure on our finite agricultural land, SI is important both for local and wider scale biodiversity outcomes. Our 'no additional production' scenarios, increased farm-scale biodiversity, but without increasing production. Increasing demand for food would therefore have to be met by intensification of other farmland and/or generating new farmland (see Fischer *et al.* 2014), with potentially negative biodiversity implications at a wider scale. Our 'additional production' scenarios that achieved SI would not only benefit local biodiversity but also reduce production pressure on land elsewhere. Ultimately, SI solutions will be important in allowing biodiversity to be increased both on a farm scale through land-sharing, and by increasing production on existing farmland which could protect biodiversity-rich areas at the landscape scale ('land-sparing'). The land sharing/sparing debate, however, can overlook the wide range of ecosystem services agricultural land can provide (Fischer *et al.* 2014) and measures that promote biodiversity often improve landscape multi-functionality enhancing a wide-range of ecosystem services (Allan *et al.* 2015).

Potential for implementation to real systems

A key finding was the important role that crop heterogeneity played in increasing beta-diversity and production. Loss of heterogeneity (of both semi-natural and agricultural habitats) is a key driver of biodiversity declines, and its restoration represents a mitigation strategy (Benton, Vickery & Wilson 2003). Agricultural policies aimed at stemming biodiversity loss typically incentivise farmers to increase heterogeneity via agri-environment schemes (AES), which may involve in-production (land-sharing) or out-of-production (land-sparing) approaches (Batáry *et al.* 2015). Restoring semi-natural habitats in intensive agricultural landscapes typically involves removing land from production, to

329 increase spatial and structural heterogeneity (Ovenden, Swash & Smallshire 1998). Our results
330 suggest that incentivising farmers to increase heterogeneity of their productive land could lead to
331 biodiversity gains whilst increasing production. At a regional level, that could involve a coordinated
332 approach to produce crops that have high complementarity and are not already dominant.
333
334 In introducing a crop diversification component within its compulsory greening measures (EU
335 Regulation 1307/2013), the EU's 2014 Common Agricultural Policy reform may represent a step in
336 the right direction. However, 'diversification' in this legislation narrowly focuses on compositional
337 heterogeneity, bringing its effectiveness under question. Josefsson et al. (2017) found that crop
338 composition *per se* did not influence bird species richness, but structural crop heterogeneity did
339 have a positive effect. For such measures to make a real impact, they may need to go further and be
340 backed-up by a political will to diversify farmland. Some AES are piloting a cluster farming approach
341 to provide landscape-scale benefits (e.g. Natural England 2017); SI strategies could also potentially
342 operate at a multi-farm scale.
343
344 With SI goals aiming to increase production and thus income, appropriate knowledge exchange may
345 enhance uptake of SI-focussed management without additional incentives, although in the UK some
346 innovative farms seen to achieve SI outcomes had relied on AES subsidies to enhance biodiversity
347 (Firbank *et al.* 2013). Improving links between researchers, advisors and farmers are identified as
348 important for uptake of SI practices in developing countries (Pretty, Toulmin & Williams 2011).
349 Promoting biodiversity can have direct economic benefits through enhancing ecosystem services
350 such as natural pest control and pollination, potentially contributing to SI via ecological
351 intensification (Tscharntke *et al.* 2012). Promoting the potential economic benefits of enhancing
352 biodiversity may also be important (Pywell *et al.* 2015). For example, insect pollination can increase

353 yield of oilseed rape (Bommarco, Marini & Vaissière 2012) while the presence of natural enemies of
354 aphids can increase yield of barley (Östman, Ekbom & Bengtsson 2003).
355
356 Incentivising heterogeneity requires careful consideration of costs and benefits. Crop heterogeneity
357 may promote economic sustainability, providing insurance against unpredictable growing conditions,
358 pest or disease outbreaks, or market variability (Garnett *et al.* 2013). In our home-grown system, the
359 production of additional crops not directly required for *in situ* cattle, requires additional market
360 engagement, but markets can be unpredictable. Growing new crops may also incur direct costs, such
361 as those for new machinery or alternative agrochemicals. Our simple measure of increased
362 production was solely based on the energetic yield of a crop, while demand depends on other socio-
363 economic drivers such as available income and consumer behaviour (Valin *et al.* 2014).
364
365 Simultaneously enhancing biodiversity and production clearly requires incorporating complex
366 information derived from disparate sources. For example, here we integrated biodiversity data from
367 both published literature, dietary modelling that combines nutritional equations to determine
368 protein and energy requirements, and finally crop production data. Farmers are familiar with a range
369 of decision-support tools, such as nutrition models, yield estimates, agronomist advice and
370 guidelines on incentives for AES. Comprehensive biodiversity datasets that evaluate biodiversity
371 across habitats are, however, often only available as summary information in scientific publications
372 (e.g. Cole *et al.* 2017). This could make it difficult for farmers to adequately consider biodiversity in
373 decision-making processes. There is great potential for existing monitoring schemes (e.g. UK's
374 Breeding Bird Survey: https://www.bto.org/volunteer-surveys/bbs; UK's Butterfly Monitoring
375 Scheme: http://www.ukbms.org/) to assist in the collection of comprehensive biodiversity data
376 across taxa and land covers, but more fundamental is providing the resultant data to land managers
377 in a usable format. Our framework highlights the potential for current databases (e.g. crop

productivity and land-cover/biodiversity) to be integrated to create a SI decision-support tool. In addition to including production and biodiversity outcomes, such a tool could also include other environmental (e.g. reducing greenhouse gas emissions, mitigating diffuse pollution) and agronomic (e.g. reduced agrochemical applications) benefits.

Simulation studies are important first steps in assessing potential SI outcomes (e.g. Simons & Weisser 2017). Ultimately, however, model predictions require trialling in real landscapes. While simulated studies provide data-driven working hypotheses, they have inevitable limitations. For example, we considered compositional but not configurational heterogeneity, which can drive biodiversity patterns in birds (Hiron *et al.* 2015). In addition, data constraints meant that to represent high-biodiversity, low-input habitats in our 'no additional production' scenarios we used data from historical extensively grazed grasslands rather than potentially more appropriate AES prescriptions (i.e. habitats formed from previously cultivated land that have not had a history of low-input management). Extensive grasslands support unique species assemblages, and it is unlikely that AES prescriptions (e.g. species-rich field margins) would reach the potential biodiversity value of such grasslands (Downie *et al.* 1999; Wilson *et al.* 2003). Our 'no additional production scenarios' may therefore over-estimate biodiversity gains. Agri-environment interventions can, however, not only increase ecological connectivity and provide ecological contrast that enhances resource diversity, stability and availability supporting a wider suite of species (Batáry *et al.* 2015; Cole *et al.* 2017), but can also enhance biodiversity-dependant ecosystem services thus benefitting production (Pywell *et al.* 2015). Landscape trials of scenarios could elucidate effects of configurational, structural and compositional heterogeneity, and test biodiversity benefits of AES prescriptions against our assumptions using historical extensively grazed habitats.

A key reason for investigating potential for SI outcomes in a home-grown dairy system was the closed nature of the system, whereby the land required to feed the herd is *in situ*. Feeding housed cattle is a common dairy production system (e.g. 55% of UK dairy farms house cattle year-round, a percentage which is growing - March *et al.* 2014) but housed cattle can be fed on home-grown or bought-in feeds, or a combination. Home-grown systems represent one end of a spectrum of possible feeding systems for housed dairy cattle (Roberts & March 2014), one which is fully self-sufficient in terms of feeds. While data on the extent of feed self-sufficiency of farms are not widely available, it can be regionally high (e.g. averaging 79-85% in W France: Brocard *et al.* 2016). Higher self-sufficiency of feeds is seen to reduce consumption of non-renewable energy (i.e. reduced transport of bought-in feeds) and raise nutrient efficiency (Gaudino *et al.* 2018). In Europe, producing local sources of protein for dairy cattle is likely to reduce reliance on imported soybean *Glycine max* imports, associated with high greenhouse gas emissions (Hörtenhuber, Lindenthal & Zollitsch 2011), while programs have been established aiming to increase levels of dairy feed self-sufficiency (Ineichen *et al.* 2014). Our simulations demonstrate a further potential environmental benefit of home-grown systems - that composition of feed crops could be managed locally to provide simultaneous productivity and biodiversity gains.

Conclusions

Achieving SI is an important but ambitious aim (Tilman *et al.* 2011; Garnett *et al.* 2013). Our study combined biodiversity data, yield data and nutrition models to find routes to simultaneous increases in production and biodiversity in a home-grown dairy system. This was optimised by maximising land-efficiency and targeting additional production to a land cover that had relatively high species richness (alpha-diversity), and complemented species in existing crops (beta-diversity). This highlights the importance of integrating agronomic efficiency, land cover heterogeneity and species richness/complementarity of both productive and non-productive land covers within an SI

framework. Where comprehensive biodiversity/production data exist, our framework could be adaptable to other taxonomic groups, production systems and regions. Agri-environment policy should focus not only on increasing the quality and heterogeneity of semi-natural habitats, but also on enhancing agricultural efficiency and the complementarity and heterogeneity of productive land covers. Through developing a framework that integrates crop productivity and biodiversity modelling to seek optimal production-biodiversity scenarios, this study presents a route to identify key drivers of production and biodiversity gain, a key goal of SI, that may be applicable at a wider scale.

AUTHORS' CONTRIBUTIONS

All authors developed the initial concepts and MAL designed test scenarios. PJCW conducted analyses and LJC sourced the datasets. PJCW and LJC wrote the initial draft and all authors contributed to revisions and gave final approval for publication.

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DATA ACCESSIBILITY

Data available from Edinburgh Napier University Repository. DOI: 10.17869/enu.2019.1497843 (White *et al.* 2019).

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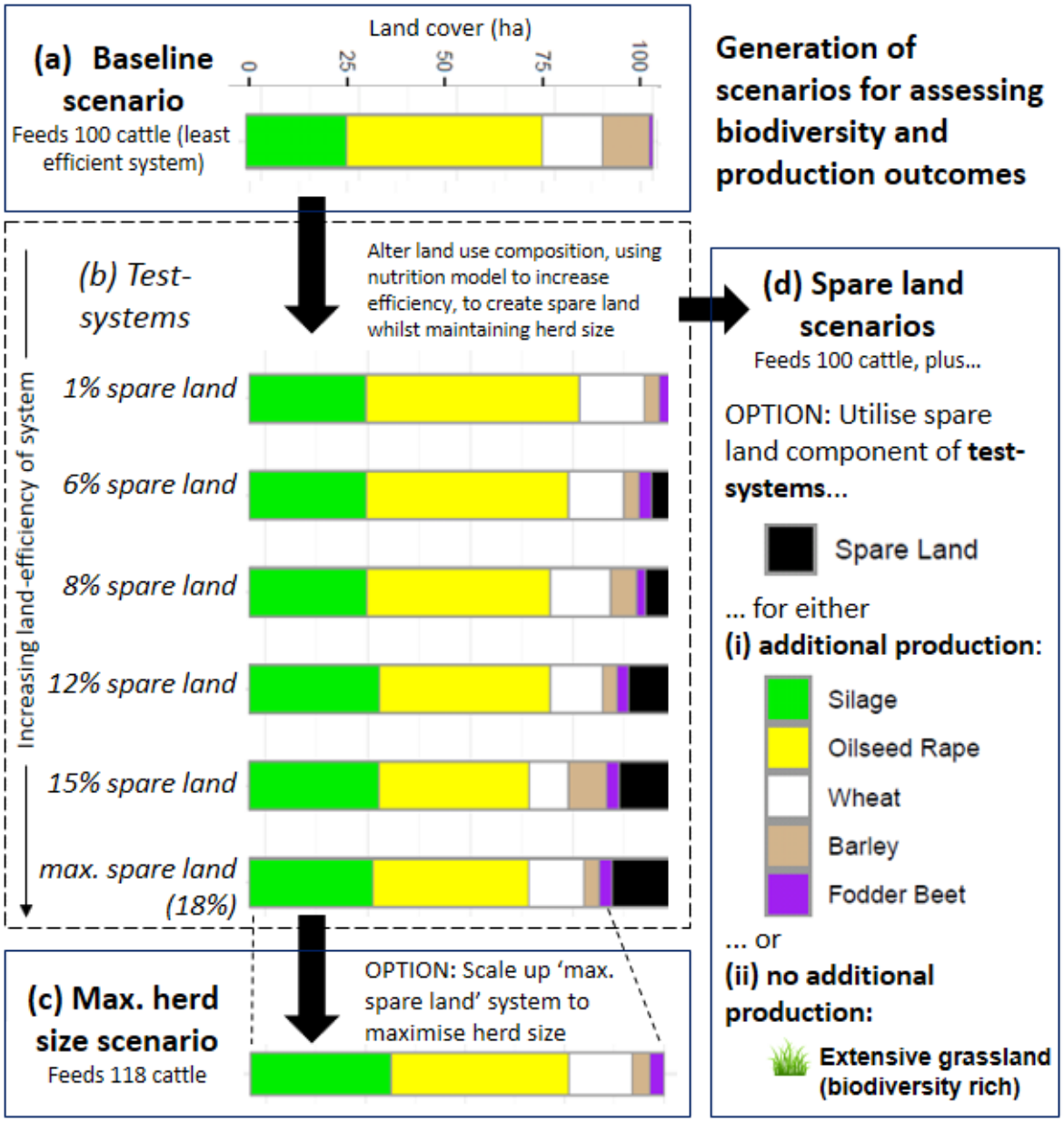

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585 **FIGURE 1. The generation of 38 dairy system scenarios. The feed ration requiring the most land**

586 **(103.8 ha) was designated the baseline scenario. Altering the farm's crop composition generated**

587 **spare land (b), without reducing herd size or milk yield (see text). This spare land could be used to**

588 **(c) scale up the system to maximise herd size, (d i) for additional production or (d ii) for**

589 **maximising biodiversity through addition of extensive grassland (no additional production).**

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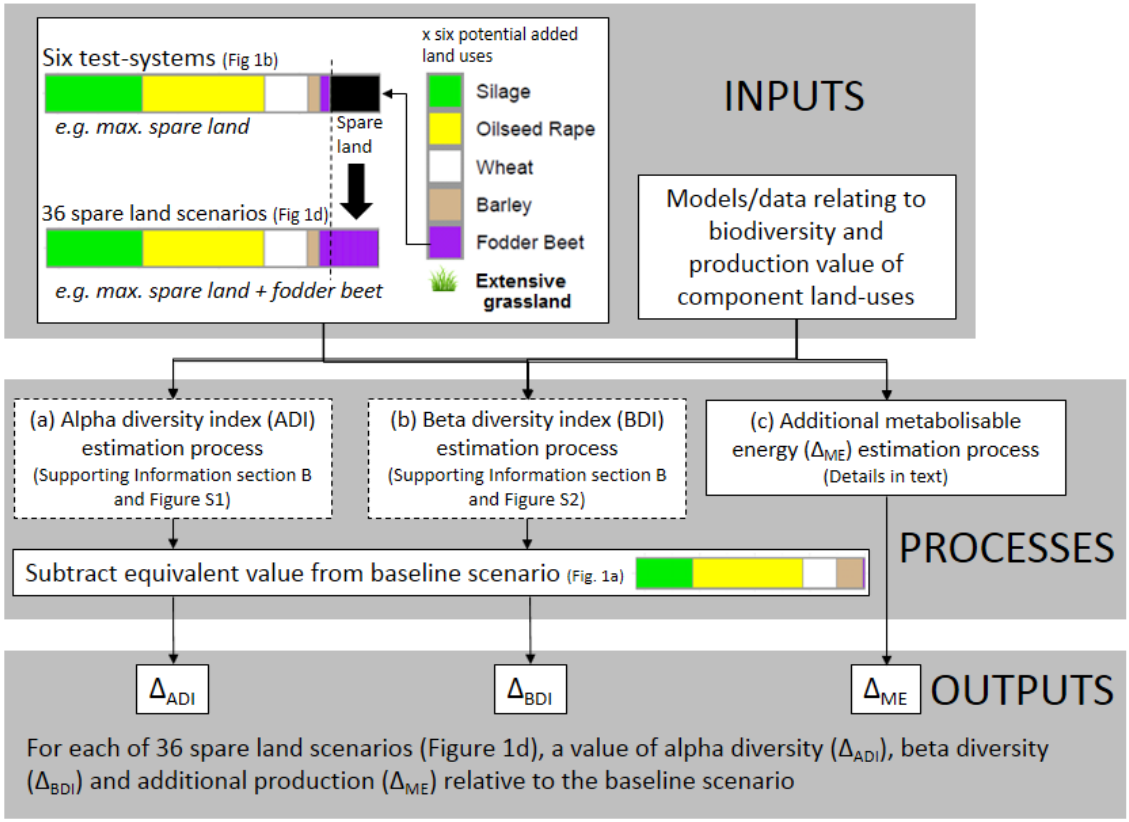
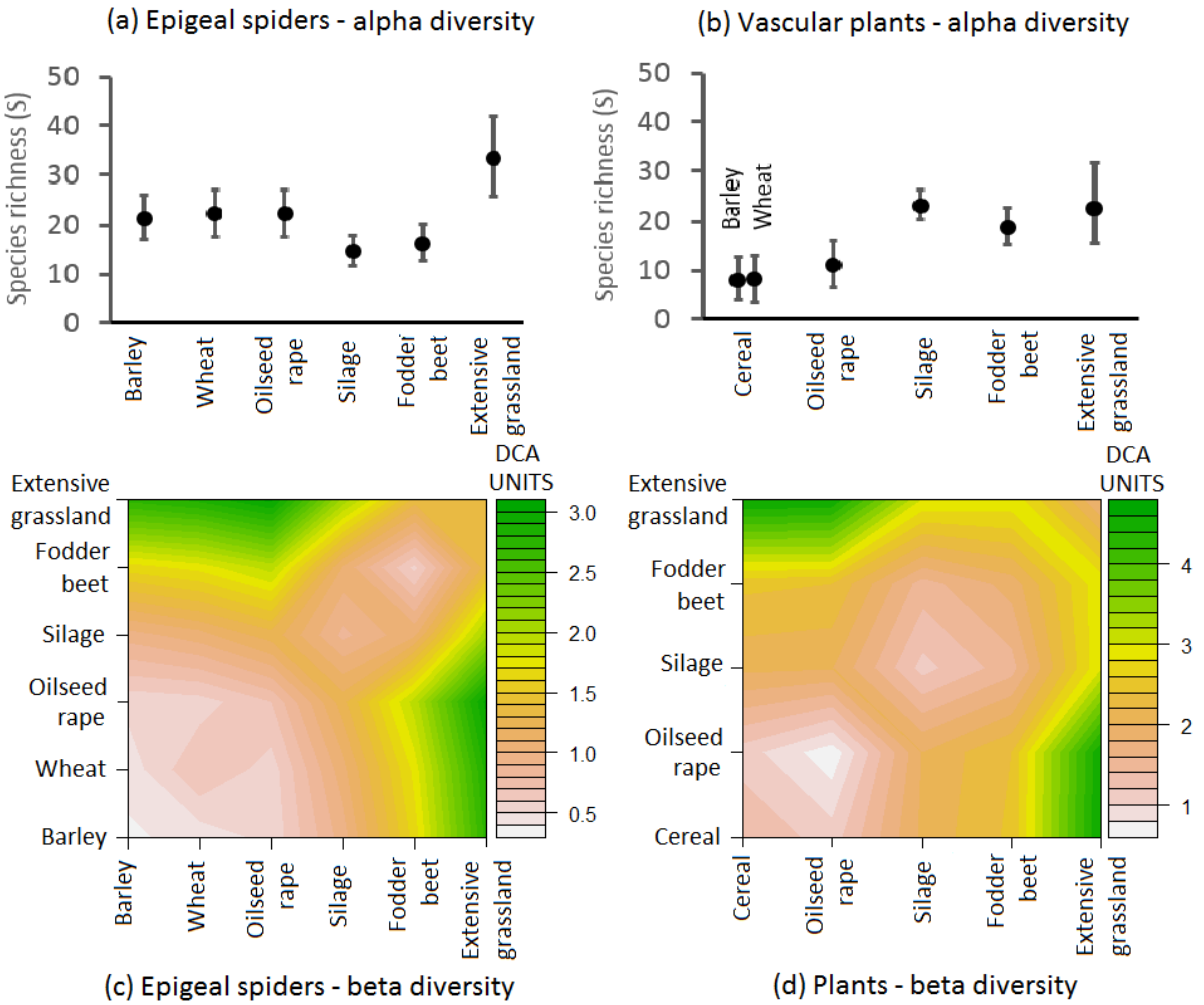


FIGURE 2. Workflow for producing estimates of alpha-diversity, beta-diversity and additional production for the 36 spare land scenarios (Fig. 1d), relative to the baseline scenario (Fig. 1a). Further information is provided in the text. More detailed information on the estimation of (a) alpha-diversity and (b) beta-diversity indices (shown in dashed boxes), with sub-workflows and model code, are provided in Appendices S2-S4.



598
599 **FIGURE 3. Alpha and beta-diversity estimates for plants and epigeal spiders for each land-use from**
600 **datasets in Downie *et al.* (1999) and Wilson *et al.* (2003): (a)-(b) estimated field-scale species**
601 **richness (bars representing upper and lower 95% confidence limits); (c)-(d) contour plots**
602 **representing mean detrended correspondence analysis (DCA) distances between and within land-**
603 **uses (a measure of beta-diversity). Land-uses are ordered to minimise DCA distances across the**
604 **primary and secondary diagonals in (c)-(d).**
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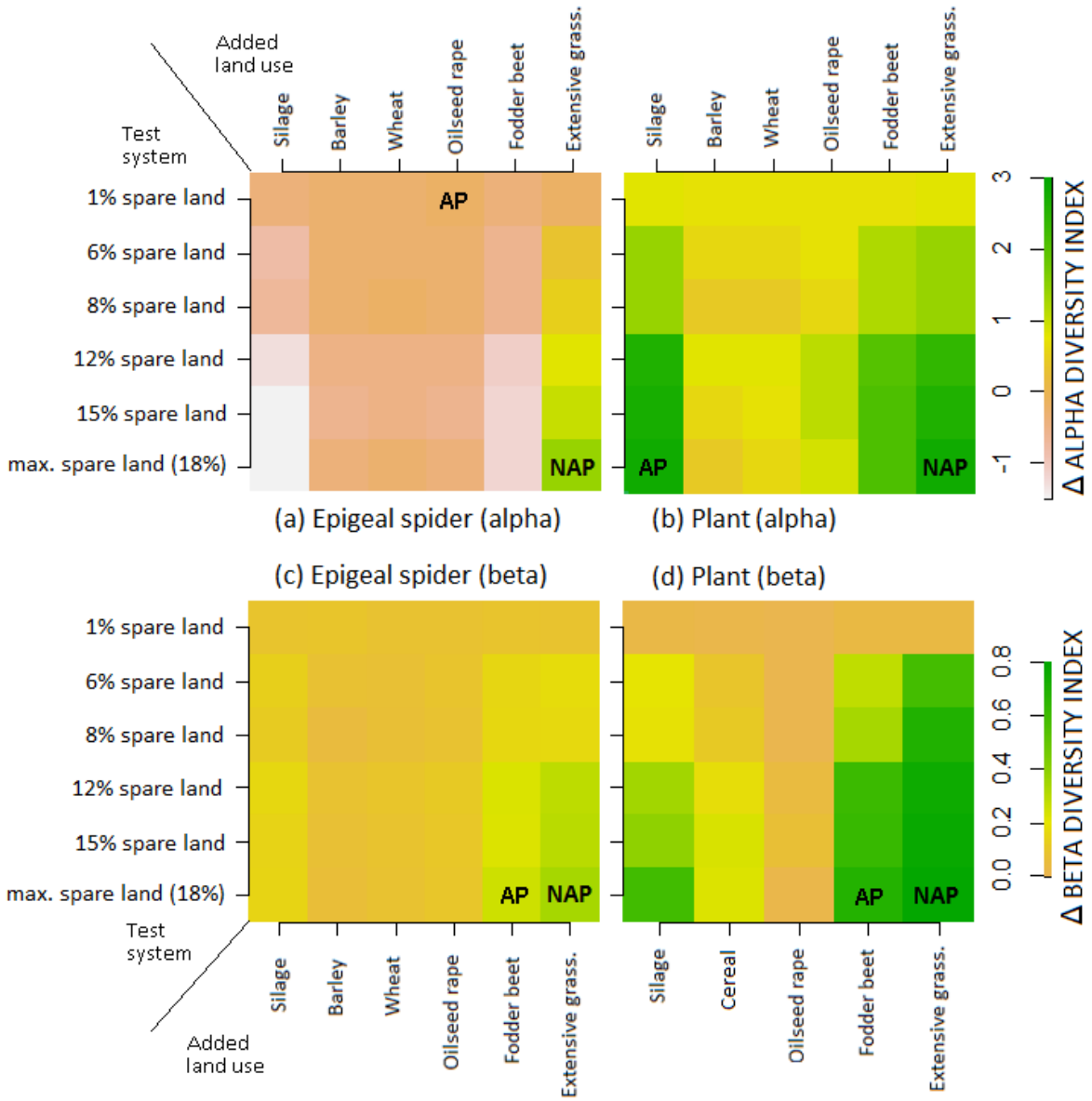

FIGURE 4. The (a)-(b) change in alpha-diversity index (Δ_{ADI}) and (c)-(d) change in beta-diversity index (Δ_{BDI}) for 36 spare-land scenarios relative to the baseline scenario, for spiders and plants. Each spare-land scenario is defined by a test-system (y-axis) and a land-use that is added (x-axis) to the spare land component of that test-system (Fig. 1). For each plot, the scenario marked 'AP' represents the highest value for an 'additional production' scenario, and 'NAP' represents the highest value for a 'no additional production' scenario (Fig. 1d). Derivation of alpha- and beta-diversity indices from field-scale biodiversity data are described in the text.

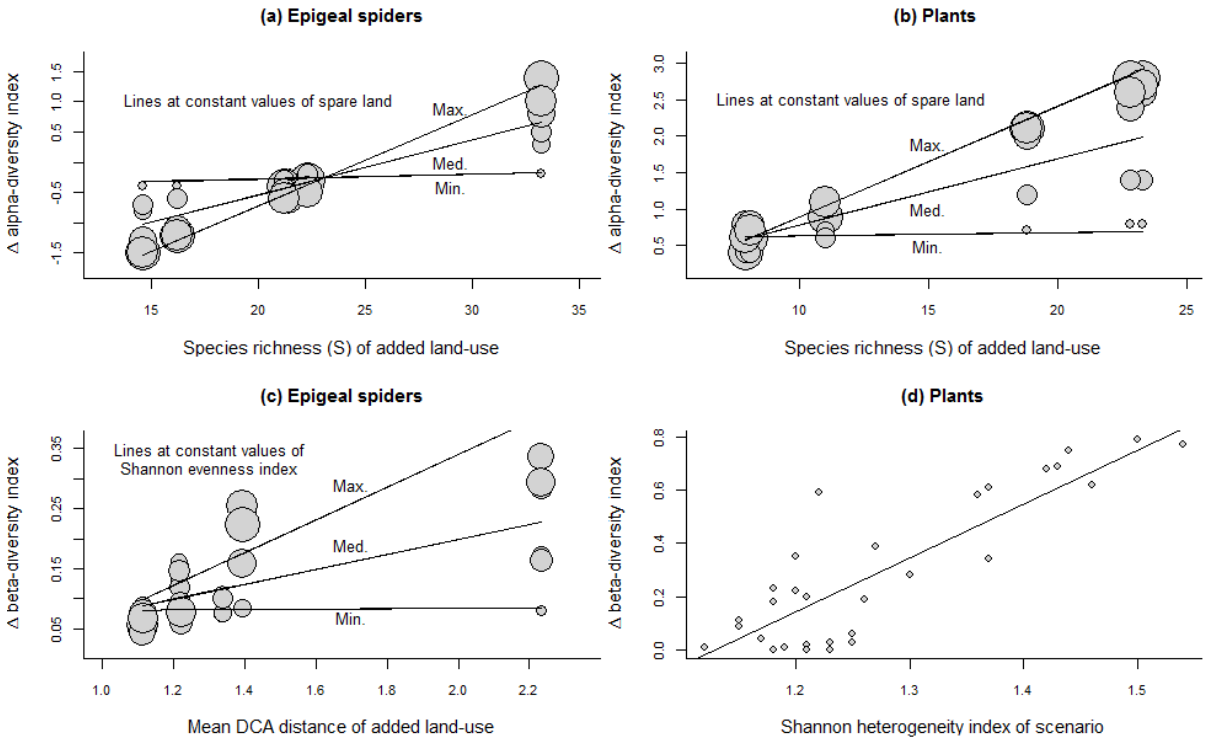

FIGURE 5. Plots of the best AIC_c models presented in Table 2. For presentation purposes, on (a)-(c) circle size is indicative of the relative value on a z-axis (in [a] and [b] this represents the quantity of spare land available, and in [c] this represents the natural logarithm of the Shannon land-use evenness index of the scenario). Interactions on (a)-(c) have been indicated by selecting three constant values on the z-axis (the minimum, median and maximum), and showing the cross-section of the modelled plane at that value. Derivation of alpha- and beta-diversity indices from field-scale biodiversity data are described in the text.

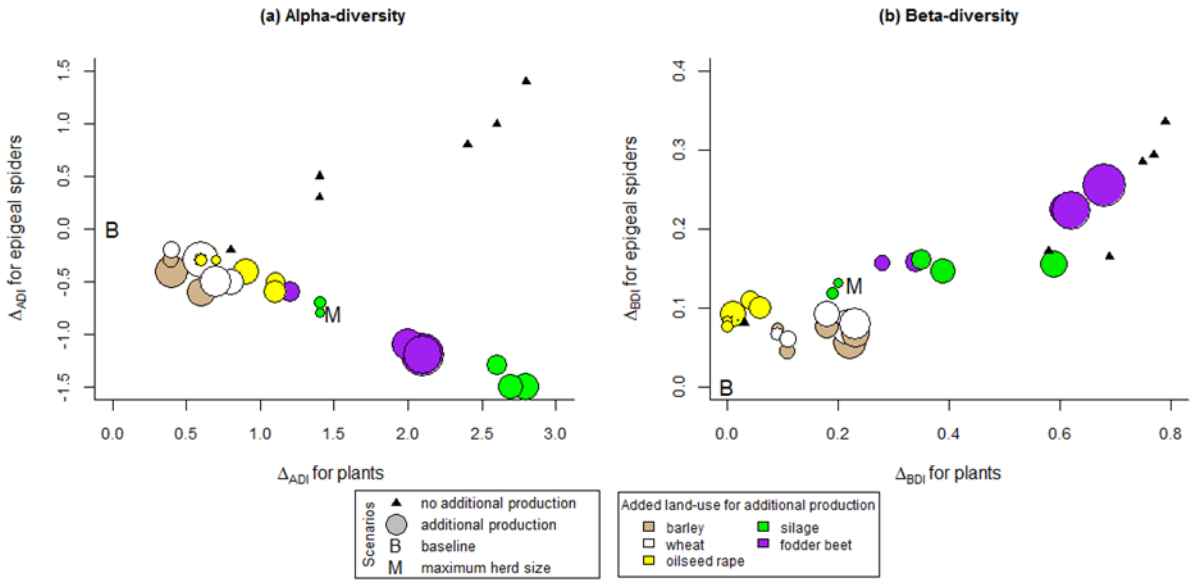

FIGURE 6. Plots of diversity indices, (a) Δ_{ADI} and (b) Δ_{BDI}, for spiders and plants for spare-land scenarios (Fig. 1d). For 'additional production' scenarios, the estimated additional metabolisable energy (ME) is indicated by the relative radius of the circle (max for both plots is 1,783 GJ dry matter). The 'no additional production' scenarios (addition of extensive grassland to spare land) do not have additional ME so are shown by ▲. For comparison, the baseline scenario (Fig. 1a) and max. herd size scenario (Fig. 1c) are shown by 'B' and 'M' respectively. Derivation of alpha- and beta-diversity indices from field-scale biodiversity data are described in the text.

Table 1. Fixed effects included in model sets for investigating drivers of alpha (Δ_{ADI}) and beta (Δ_{BDI}) diversity indices of scenarios relative to the baseline scenario (Fig. 1a).

Code	Description	In model set: Δ_{ADI}	In model set: Δ_{BDI}
SPARE	Area (ha) of spare land available in the test-system	X	X
HETER	Shannon heterogeneity index of the land-use composition of the scenario	X	X
EVEN	Shannon evenness index of the land-use composition of the scenario	X	X
LANDUSES	Number of land-uses within the scenario (5 or 6)	X	X
COVER	Initial area (ha) in the test-system for the land cover replacing the spare land component	X	X
RICHNESS	Mean estimated S of the given taxa of the land-use replacing the spare land component (see Fig. 2)	X	
DCA	Mean Euclidean detrended correspondence analysis distance for the given taxa of the land-use replacing the spare-land component against each other land-use (including itself)		X

Table 2. Ranking table for models of changes in alpha- and beta-diversity from the baseline scenario, with the number of model parameters (k), small-sample AIC (AICc), difference in AICc from the lowest AICc value (ΔAICc) and the Akaike's weight (w_i) of each. Model terms are defined in Table 1. [NULL] = null model. The form *x*z* indicates an interaction between *x* and *z*. The confidence set (summed $w_i \geq 0.90$) is in bold. For brevity, only the confidence set, the model immediately outside it, and the null model are displayed. The sample size for each model was 36, representing the 36 spare land scenarios. Parameter estimates and SEs for all confidence set models are in Supporting Information Table S2.

	Epigeal spiders					Plants				
DIVERSITY	MODEL	k	AICc	ΔAICc	w_i	MODEL	k	AICc	ΔAICc	w_i
Alpha (Δ_{ADI})	**RICH*SPARE**	**4**	**-50.4**	**0.0**	**>0.99**	**RICH*SPARE**	**4**	**-46.3**	**0.0**	**>0.99**
	RICH*EVEN	4	20.0	70.4	<0.01	RICH*EVEN	4	50.3	96.6	<0.01
	....10 models…					…11 models…				
	[NULL]	1	50.5	100.9	<0.01	[NULL]	1	69.4	115.7	<0.01
	… 4 models…					…3 models…				
Beta (Δ_{BDI})	**DCA*EVEN**	**4**	**-117.3**	**0.0**	**0.87**	**HETER**	**2**	**-25.8**	**0.0**	**0.71**
	DCA*HETER	**4**	**-112.4**	**4.8**	**0.08**	**EVEN**	**2**	**-22.33**	**3.5**	**0.13**
	DCA*SPARE	4	-111.6	5.6	0.05	**DCA+HETER**	**3**	**-20.57**	**5.2**	**0.05**
	…4 models…					**DCA+EVEN**	**3**	**-20.53**	**5.3**	**0.05**
	[NULL]	1	-93.22	24.0	<0.01	DCA*EVEN	4	-19.9	5.9	0.04
	… 9 models…					…3 models…				
						[NULL]	1	-3.5	22.3	<0.01
						…8 models…				